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Weighting in LCA Based on Ecotaxes

Development of a Mid-point Method and Experiences from Case Studies

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Abstract

Goal, Scope and Background. The weighting phase in Life Cycle Assessment (LCA) is and has always been a controversial issue, partly because this element requires the incorporation of social, political and ethical values. Despite the controversies, weighting is widely used in practise. In this paper we will present an approach for monetisation of environmental impacts which is based on the consistent use of ecotaxes and fees in Sweden as a basis for the economic values. The idea behind this approach is that taxes and fees are expressions of the values society places on resource uses and emissions. An underlying assumption for this is that the decisions taken by policy-makers are reflecting societal values thus reflecting a positive view of representative democracy.

Methods. In the method a number of different ecotaxes are used. In many cases they can directly be used as valuation weighting factors, an example is the CO_2-tax that can be used as a valuation of CO_2-emissions. In some cases, a calculation has to be made in order to derive a weighting factor. An example of this is the tax on nitrogen fertilisers which can be recalculated to an emission of nitrogen which can be used as a weighting factor for nitrogen emissions. The valuation weighting factors can be connected to characterisation methods in the normal LCA practise. We have often used the Ecotax method in parallel to other weighting methods such as the Ecoindicator and EPS methods and the results are compared.

Results and Discussion. A new set of weighting factors has been developed which has been used in case studies. It is interesting to note that the Ecotax method is able to identify different environmental problems as the most important ones in different case studies. In some cases, the Ecotax method has identified some interventions as the most important ones which lack weighting factors in other weighting methods. The midpoint-endpoint debate in the LCA literature has often centred on different types of uncertainties. Sometimes it is claimed that an advantage with having an endpoint approach is that the weighting would be easier and less uncertain. Here we are however suggesting a mid-point weighting method that we claim are no less uncertain than other often used weighting methods based on a damage assessment. This paper can therefore be seen as a discussion paper also in the midpoint-endpoint debate.

Conclusion and Recommendation. The Ecotax method is ready to use. It should be further updated and developed as taxes are changed and new characterisation methods are developed. The method is not only relevant for LCA but also for other environmental systems analysis. The Ecotax method has also been used as a valuation method for Cost-Benefit Analysis (CBA), Life Cycle Costing (LCC) and within the context of a Strategic Environmental Assessment (SEA).

Keywords: Cost-benefit assessment; impact assessment; LCA; LCC; strategic environmental assessment; valuation; weighting

Introduction

A large number of tools for assessing environmental impacts are available (Wrisberg et al. 2002, Finnveden and Moberg 2005). Examples include Cost-Benefit Analysis (CBA), Life Cycle Assessment (LCA), Life Cycle Costing (LCC) and Strategic Environmental Assessment (SEA). In many of these tools, a valuation or a weighting step is or can be used.

LCA is divided in several phases and the Life Cycle Impact Assessment (LCIA) phase is divided into several elements according to the ISO standard (ISO 1997 and 2000). In the characterization, indicator results are calculated within impact categories. In the normalisation element, the results from the characterisation are related to some reference values. In the weighting step results from the characterization or normalization are converted using numerical factors based on value choices; it may include aggregation across impact categories of the weighted results (ibid). It can be noted that, according to this definition, weighting is a quantitative process. Normalisation and weighting are described as optional elements which can be used depending on the goal and scope of the LCA, although weighting is not allowed for comparative assertions disclosed to the public (ibid).

Of significance is that the last phase of an LCA is Interpretation where conclusions are drawn and recommendations are made (ISO 1997). The results from the weighting are thus not necessarily the final conclusions drawn from the overall LCA study, nor the only factors considered in a decision. Although the ISO standards provide a framework and

a terminology for LCA, it is clear that there is on-going development including different methodological choices. It is also often noted that the scientific and methodological framework for LCIA is still being developed (ISO, 2000) and has been under discussion in several international forums (e.g. Bare et al. 1999 and 2000, Jolliet et al 2004, Udo de Haes 1996, Udo de Haes and Wrisberg 1997, Udo de Haes et al. 1999a,b and 2002a,b). Recent reviews of LCIA methods include Udo de Haes et al (2002a) and Pennington et al. (2004).

The weighting phase in LCA is and has always been a controversial issue, partly because this element requires the incorporation of social, political and ethical values, (c.f. Finnveden 1997). Not only are there values involved when choosing weighting factors, but also when choosing which type of weighting method to use, and also in the choice whether to use a weighting method at all (ibid.). Despite the controversies, weighting is widely used in practice (e.g. Hansen 1999). It is therefore important to critically review methods and data used. Evaluating weighting methods is however difficult since the values involved are difficult to identify and evaluate. However, all weighting methods use data and methods taken from different scientific disciplines that can and should be evaluated, and the value choices can be identified and clarified.

1 Different Types of Weighting Methods

Methods for weighting can be classified in different ways (Finnveden et al. 2002):

- Often a distinction is made between panel methods and monetisation methods. In panel methods, a group of people are asked about their values. The common aspect of monetisation methods is that the values are expressed in a monetary measure. Sometimes also a third group is mentioned, distance-to-target methods, where the weighting factors are calculated as a function of some type of target value. However, it may be questioned if distance-to-target methods are weighting methods since the different targets are not weighted against each other (Finnveden et al. 2002).
- A distinction can be made between methods based on stated and revealed preferences. In methods based on stated preferences, people are asked about their preferences and this information is used. All panel methods and some monetisation methods are based on stated preferences. In methods based on revealed preferences, preferences revealed in different situations are used to calculate weighting factors. Some monetisation methods are based on revealed preferences.
- Finally a distinction can be made between mid-point methods and end-point or damage methods. This distinction is based on the cause-effect chain or environmental mechanism of environmental problems (Finnveden et al. 1992, Bare et al. 2000, Udo de Haes et al. 2002a). The cause-effect chain starts with some emissions or other types of activities which lead to what can be called primary changes in the environment. These primary changes lead to secondary and tertiary changes etc. Early in the cause-effect chain are often chemical and physical changes, e.g. changes in concentrations in the atmosphere or changes in infrared radiation (in the case of climate change). Later in the cause-effect chain are often biological changes, e.g. changes in ecosystems or human health. Methods that are based on changes early in the cause-effect chain may be called mid-point methods in contrast to end-point or damage methods which are based on changes later in the cause-effect mechanism.

Monetisation methods are here used as an umbrella term for all methods which have a monetary measure as the unit for weighting factors. Environmental economists make a distinction between different types of 'economic values' relating to natural environments. The terminology is not completely agreed upon, the following is however based on Turner et al. (1994). A first distinction is made between use values and non-use values. The use values include both direct and indirect use values. An example of a direct use value is the timber value of a forest. The indirect use value includes the recreation value of the forest, the value of carbon fixation etc. (ibid.). Non-use values are non-instrumental values that are attributed to objects without the direct intention of actually using them (ibid.). Such values include existence, bequest and option use values (ibid). The total economic value is the sum of the use and non-use values (ibid.).

There are a large number of different approaches for monetising environmental impacts. The classification here is based on Turner et al (1994), Finnveden (1999), and Bockstael et al. (2000):

1. Methods based on willingness to pay
1.1 Individual's revealed preferences
1.2 Individual's stated preferences
1.3 Willingness to pay revealed through societal decisions
2. Methods not based on willingness to pay

The first distinction is made between methods that are based on willingness to pay and methods that are not (no distinction is here made between 'willingness-to-pay' and 'willingness-to-accept'). Methods based on willingness-to-pay measure an 'economic value' (Bockstael et al. 2000). Methods based on individual's revealed preferences are assuming that people reveal their preferences in market behaviour. The revealed preferences are normally only related to the use values, and sometimes only the direct use value. Direct use values can often be derived from actual market prices, e.g. the market price of timber. In addition, some indirect use values may be derived from market values, for example through the travel cost method and hedonic pricing methods.

Non-use values can normally not be derived from revealed preferences (Turner et al. 1994). The contingent valuation method (CVM) bypasses the need to refer to market prices by asking individuals explicitly to place values upon environmental assets (ibid.). Because of this, the CVM is often referred to as an expressed or stated preference method (ibid.). CV methods have been used extensively and there are guidelines developed for them (Carson 2000). As an alternative method to the use of open ended bidding questions

or dichotomous choices of referendum questions conjoint analysis has been developed (Farber and Griner 2000).

A willingness to pay may be derived from political and governmental decisions. In these methods, it is assumed that meaningful information on environmental values can be derived from political and governmental decisions. One way of deriving a 'societal price' or a collectively revealed preference is to study society's efforts to avoid impacts. An example may be the costs of reducing emissions to a decided emission limit. The marginal cost for removing the pollutant to the emission limit can be seen as the monetary value the society puts on the pollutant. These costs may be called prevention costs or abatement costs. Yet, another way of deriving a 'societal price' is to look at 'green taxes'. If there are taxes on emissions, these taxes may be seen as an expression of the value the society is putting on the emissions. However, the tax level is influenced also by other aspects than the valuation of the environmental effect of the emission. This is because introduction of a tax will not only lead to reduced emissions but also other effects. Examples include effects on competitiveness of firms within the country if they have to pay taxes. That is one reason why there often are tax exemptions for industries. Since these effects often are used for arguments for lower taxes, it may be argued that tax exemptions can be regarded as subsidies and the highest tax level can be regarded as a better approximation of the societal valuation of the environmental effects of the emission.

There are also a number of monetization methods that are not based on willingness-to-pay. Methods that are not based on a willingness-to-pay do not measure the 'economic value' (Bockstael et al. 2000). They are often based on an estimation of a cost to do something, however if it is not clear that somebody is willing to pay this cost, it is not a measure of a willingness-to-pay.

Since different monetization methods cover different types of economic values (i.e. different use and non-use values), different methods should result in different results, and they do. For example, the total economic value, as measured by the contingent valuation method, is in some cases an order of magnitude larger than the economic value derived from market valuations (KI 1998). This may be explained by the non-use values that are included by CVM. Just because something is expressed in monetary terms does not make it immediately comparable or additive to another measure in the same unit using a different method (cf. Bockstael et al. 2000). If a monetization weighting method is used, the same method should therefore ideally be used to derive all economic values within the weighting method. Care should also be taken when comparing the results from a monetization weighting method and other types of costs.

A number of different weighting methods have been developed for LCA and are being used e.g. the Ecoindicator (Goedkopp and Spriensmaa 1999) the EPS method (Steen 1999) the LIME method (Itsubo et al. 2004) and a Chinese method (Wu et al. 2005). A number of different approaches were reviewed by the SETAC working group on LCIA (Finnveden et al. 2002). They used a system of criteria for evaluating different approaches and concluded that there is currently no method that fulfils all relevant criteria.

Aim of this study. The aim of this paper is to describe the Ecotax method developed by Johansson (1999) and Eldh (2003) and show how it can be used in case studies. The Ecotax method is consistently based on taxes and fees on emissions and resources uses in Sweden. The taxes and fees are taken as expressions of the value society puts on the environmental effects of the emissions or resource uses.

2 Developing Ecotax 2002

2.1 Principles

The development of the Ecotax method takes place in several steps:

1. Choice of impact categories. The taxes will be used as weighting factors to weigh different impact categories against each other. A choice of impact categories thus has to be made. Here we use the Base line set of the Dutch guideline on LCA (Guinée 2002) as implemented in the Simapro software except for resources where the thermodynamic approach is used (Finnveden and Östlund 1997).

2. Identification of suitable taxes and fees. Eurostat and the OECD have elaborated a definition of environmental taxes that has been accepted by the member states. This makes comparative studies possible between different countries in terms of tax structure, tax base, revenues, etc. (SCB 2000). A list of environmental taxes can be found in the Swedish environmental protection agency's publication (Naturvårdsverket 2001).

3. Connecting taxes and impact categories. In some cases there is a direct connection, for example a carbon dioxide tax can be used as a weighting factor for carbon dioxide-equivalents for the impact category climate change. In other cases, the connection is more indirect and recalculations have to be made.

In some cases there are tax exemptions or subsidies. For example, in Sweden, industries pay a lower carbon dioxide tax than households. In these cases we use the higher tax level as a measure of the societal valuation of the environmental effects and regard the other levels as subsidies. In other cases, there is a tax or fee on emissions from some sources but not other from other sources, for example there is no carbon dioxide tax from production of electricity. Again, we regard these as exemptions and use the higher level as the relevant measure.

In some cases it is possible to calculate weighting factors for several interventions related to the same impact category. In such cases we use the range between the highest and the lowest as a measure of uncertainty in the values society puts on different environmental impacts. We will develop two sets of weighting factors called the min and the max versions in which the minimum and maximum values are consistently chosen.

2.2 Taxes and impact categories

All data on taxes and weighting factors are given in SEK. For order of magnitude calculations 10 SEK equals approximately 1 euro. The results are summarised in **Table 1** and **Table 2**.

Abiotic resources. Energy taxes are charged for fuels used for heating purposes or for operating motors. The tax rate is not directly related to the energy content in different fuels. Fuel used in commercial traffic on water, special industrial branches, rail traffic and the kerosene used in aviation are exempt from tax or at least exempt from part of it. The European Commission has in collaboration with the member states established a minimum excise duty (European Commission 2002). These rates are however far below the Swedish national rates.

The weighting factor for abiotic resources is based on the energy content in fossil fuels and the characterisation reference used is MJ. The excise duty rates were taken from (Fiscal Affairs Department 2002). The fuel with the highest tax per MJ is used for deriving the maximum value. The highest tax was the tax on petrol other than environmental class 1 or 2, and this gives a maximum value of 0.15 SEK/MJ. Liquefied petroleum gas or natural gas used for propulsion of motor driven vehicles, vessels, and aircraft were not taxed at all and therefore a minimum value is set to zero.

There is also a tax on natural gravel which can be used as a basis for the valuation of natural resources. If this tax is used as a basis for the valuation, the results will be lower than the max-values used here (Johansson 1999).

Biotic resources. Similar to the case with abiotic resources, the weighting factor for biotic resources is based on energy content in fuels. The tax on crude tall oil is used for deriving the maximum value of MJ-equivalents. Since all other bio fuels was exempt from the energy tax the minimum weighting factor is set to zero for the category biotic resources.

Table 1: Weighting factors for Ecotax derived from environmental taxes and fees in Sweden 2002 (Eldh 2003) (For order of magnitude calculations 10 SEK = 1 Euro)

Intervention	Weighting factor	Tax or fee base
Extraction		
Fossil energy	0–0.15 SEK / MJ	Tax on fossil energy
Biotic energy	0–0.069 SEK/MJ	Tax on biotic energy
Emission		
CO_2	0.63 SEK/kg	Tax on carbon content in fossil fuel recalculated to CO_2-emission
Ozone depleting substances	1,200 SEK/kg	Fee for using prohibited ozone depleting substances
Nitrogen	12 SEK/kg	Tax on nitrogen content of fertiliser recalculated assuming a leakage of 15% (tax 1.80 SEK/kg)
HC	20–200 SEK/kg	Emission fee for air traffic
Sulphur	30 SEK/kg	Tax on sulphur content in fossil fuels
Toluene	17.65–36.07 SEK/kg	Tax differentiation on petrol qualities (unleaded petrol vs. alkylate petrol)
Cadmium	30,000 SEK/kg	Tax on content of cadmium exceeding 5 g/1,000kg phosphor in fertiliser
Pesticides / Copper	20 SEK/kg	Tax on active substance in pesticides

Table 2: Weights used in minimum and maximum combinations of Ecotax 2002 (Eldh 2003) (For order of magnitude calculations 10 SEK=1 Euro)

Impact category	Combination	Weighting factor	Reference of the characterisation method (eq)	Weight of reference
Abiotic resources	Min	0 SEK/MJ	MJ	0 SEK/MJ
	Max	0.15 SEK/MJ	MJ	0.15 SEK/MJ
Biotic resources	Min	0 SEK/MJ	MJ	0 SEK/MJ
	Max	0 069 SEK/MJ	MJ	0.069 SEK / MJ
Global warming	Min/Max	0.63 SEK / kg CO_2	CO_2	0.63 SEK/kg
Depletion of stratospheric ozone	Min/Max	1,200 SEK / kg ozone depleting substance	CFC-11	1,200 SEK/kg
Photochemical oxidation	Min	20 SEK/kg HC	C_2H_2	48 SEK/kg
	Max	200 SEK/kg HC	C_2H_2	480 SEK/kg
Acidification	Min/Max	30 SEK/ kg Sulphur	1.2 SO_2	18 SEK/kg
Eutrophication	Min/Max	12 SEK/kg N	PO_4^{3-}	28.57 SEK/kg
Fresh water aquatic ecotoxicity	Min	17.65 SEK/kg Toluene	1,4-dichlorobenzen emitted to freshwater	60.86 SEK/kg
	Max	36.07 SEK/kg Toluene		124.37 SEK/kg
Marine aquatic ecotoxicity	Min	20 SEK/kg Copper	1,4-dichlorobenzen emitted to seawater	$1.333*10^{-5}$ SEK/kg
	Max	20 SEK/kg Glyphosate		0.606 SEK/kg
Terrestrial ecotoxicity	Min/Max	30,000SEK/kg Cd	1,4-dichlorobenzen emitted to agr. soil	176.47 SEK/kg
Human toxicity	Min/Max	30,000SEK/kg Cd	1,4-dichlorobenzen emitted to agr. soil	1.50 SEK/kg

Global warming. The characterisation reference for the impact category global warming is carbon dioxide. In 2002 the tax on carbon dioxide emitted from the combustion of fossil fuels was approximately 0,63 SEK / kg (Fiscal Affairs Department 2002), and this value is used as weighting factor.

Depletion of stratospheric ozone. The characterisation reference for depletion of stratospheric ozone is CFC-11. The weighting factor is based on a fee collected from those granted an exemption from the ban on using ozone-depleting substances. The fee was 1,200 SEK/kg ozone depleting substance, regardless of its ozone depletion potential. Therefore 1,200 SEK/kg is used as a weighting factor for all ozone depleting substances.

Acidification. The characterisation reference for acidification is sulphur dioxide. The weighting factor is derived from the tax on the sulphur content in fossil fuels, including peat. This tax was 30 SEK per kilogram of sulphur in the fuel (Fiscal Affairs Department 2002). Assuming that the total sulphur content is emitted as sulphur dioxide the weighting factor has been calculated to 18 SEK/kg sulphur dioxide.

Eutrophication. A tax on fertiliser was introduced in 1984 in order to reduce eutrophication of inland and coastal waters in Sweden. The tax is based on the nitrogen and cadmium content in fertilisers and was set to 1.8 SEK/kg for nitrogen. The taxation on nitrogen content of fertilisers is used as calculation base for eutrophication. The leakage of nitrogen from agricultural soil varies a lot but is here estimated to about 15 percent (Bernes 1991). This gives a value of 12 SEK/kg nitrogen leached. The characterisation reference for eutrophication is phosphate (PO_4^{3-}) in the characterisation method used (Guinée 2002).

As an alternative, the fee on NO_x which is 40 SEK/kg can be used. However, since NO_x contributes to several environmental problems (eutrophication, acidification, human toxicity and photochemical oxidation) an allocation of the fee between the impact categories has to be made. Since this is difficult we chose not to use the NO_x-fee for the impact category eutrophication.

Photochemical oxidation. As a part of the landing fee for aircraft over 9 tons there is an emission based fee that depends on a landing and take off emission certification (Luftfartsverket 1998). This certification is linked to the emissions of nitrogen oxides and hydrocarbons that the engines produce. The purpose of this fee is to get airline companies to use aircraft with better technology to reduce the emissions and to put pressure on the manufactures of aircraft engines to produce engines with lower emissions. In order to find the valuation of hydrocarbon emissions the value of the nitrogen oxide emission, which according to the nitrogen oxide fee on emissions from combustion was 40 SEK/kg, is first subtracted. By using data for emissions from some specific aeroplanes, the fee specific for hydrocarbons can then be calculated to between 20 and 200 SEK/kg HC (Eldh 2003). The characterisation reference is acetylene (C_2H_2) and the min value is 48 SEK/kg and the max value is 480 SEK/kg according to the characterisation method used (Guinée 2002).

Toxicity. The problems of emissions from leisure boats to water have for many years been under attention (Burman et al. 1997). Therefore a tax differentiation has been introduced in favour for alkylate-based petrol. The difference in tax between alkylate-based petrol and non-leaded petrol is 1,88 SEK (Regeringen 2001). Non-leaded petrol 95 octane contains 42 percent arenes (Fiscal Affairs Department 2002), 1 percent benzene and 13 percent alkenes. Corresponding percentage in alkylate-based petrol is 0.5 arenes, 0.1 benzene and 0.5 alkenes. The corresponding emissions from the alkylate-based petrol are also lower. In a study by Alin and Astnäs (2001) the emissions from the two types of petrol were compared for different outboard engines. The substances measured were isooctane, toluene, benzene, ethylbenzene, o-xylene, m/p-xylene, 1,2,3-tribenzene, 1,2,4-tribenzene and 1,2,5-tribenzene. In our calculations we treated them all as toluene and could then calculate the corresponding value for toluene (Eldh 2003). The results are between 17.65–36.07 SEK/kg depending on which engine is used. These data are then used for the fresh water aquatic ecotoxicity impact category calculated with the characterisation method used (Guinée 2002).

As noted above, the tax on fertilisers also involve a component for cadmium. The tax on the cadmium content in fertilisers was 30 SEK/g. This valuation is used for both the terrestrial ecotoxicity and human toxicity impact categories.

The tax on pesticides is based on the active substance in the pesticides and the rate was 20 SEK/kg active substance. Different pesticides have a range of different environmental impacts. The pesticide tax is therefore not very precise and a range of weighting factors can be calculated. Here we used two extreme pesticides for the marine ecotoxicological impact category: glyphosate and copper. The latter is used as the active substance in antifouling agents on boats especially in marine environments. Copper has a high marine toxicity potential while glyphosate has a relatively low marine toxicity potential. Assigning them both 20 SEK/kg results in a minimum and a maximum value for the characterisation reference, which in this case is 1,4-dichlorobenzene (Guinée 2002).

Summary. The weighting factors derived from taxes and fees are summarised in Table 1. The weighting factors for the specific impact categories are shown in Table 2. Minimum and maximum weighting factors are used in some cases indicating uncertainties in the values.

3 Results from Case Studies

The Ecotax 1999 and 2002 methods have been used in a number of case studies by us and by others (e.g. Carlsson Reich 2005, Engström et al. 2005, Bäckman et al. 2001, Finnveden et al. 2005, Hochschorner et al. 2005, Moberg et al. 2005, Nilsson et al. 2005). Here we will discuss results from some of these.

The Ecotax 2002 method was used in a case study were a proposed tax on incineration of solid waste was evaluated (Björklund et al. 2003, Nilsson et al. 2005). The study was an LCA but made in the context of a test of methods for Strategic Environmental Assessment. In parallel also the Ecoindicator hierarchist version (Goedkoop and Spriensmaa 1999) and the EPS method (Steen 1999) as implemented in the Simapro software were used. In the study the waste

management system in the year 2008 was modelled. A key methodological factor when modelling waste management systems is the time aspect which especially for landfills can be decisive (Finnveden 1999). Here we used two time perspectives: The surveyable time period (RT=0) corresponding to approximately one century and the remaining time period (RT) corresponding to a hypothetical infinite time period (Finnveden et al. 1995). The different weighting methods were applied and the question was asked, which are the most important environmental impacts according to the different methods.

For the Ecotax method, the results are dependent on the time horizon used. If long time perspectives are included, the results are dominated by emissions of metals from the ashes. If only shorter time perspectives are considered, the most important interventions in the Ecotax max version are emissions of HF to air, the use of biomass and emissions of Ni, Se and Ba to water. For the Ecotax min (RT=0) version the emissions to water are still important but supplemented with Cr, Ni, Cu to soil, emissions of CO_2 and N_2O to air, Cu, V and PAHs to water and hydrocarbons to air. Thus quite a mixture of interventions is important. This is in contrast to the same weighting methods when also the remaining time is considered. Here the results are dominated by emissions of Cu, Ni and Zn.

The results for the EPS system are quite different. The emissions of metals to water which is dominating according to the Ecotax method for the remaining time, do not even have weighting factors in the EPS-system, so the results are the same regardless of if the remaining time is considered or not. Instead the results are dominated by resource extractions and emissions contributing to climate change with some contributions from dust and PAHs to air.

For the Ecoindicator, the results are dominated by emissions of Cr during the remaining time period. If these are excluded, the most important interventions are the extractions of raw materials (crude oil and natural gas). Also for the Ecoindicator it can be noted that some interventions which are important according to the Ecotax methods lack weighting factors.

We also used the same set of methods for a study on the total Swedish agricultural production (Engström et al. 2005).The inventory was largely based on environmentally extended input-output analysis (IOA). In this study the most important environmental problems according to the Ecotax method was eutrophication followed by the use of resources and climate change. According to the EPS method the most important problem was climate change followed by abiotic resources. According to the Ecoindicator, the most important problem was the use of fossil fuels followed by emissions causing respiratory problems.

In yet another, quite different study, we used the same set of methods on a case study on a grenade (Hochschorner et al. 2005). The grenade is typically used for combat vehicles of the type often used by United Nations' peacekeeping troupes. We evaluated the grenade from an environmental point of view for two scenarios, a peace scenario where 95% of the grenades are stored and later demilitarised and the remaining 5% are used in practise, or a war scenario where all of the grenades are used. In the latter case, only the emissions of the exploding grenade were considered in the quantitative assessment of the use phase of the grenade.

In both the peace and war situations the dominating environmental impact according to the Ecotax method is the toxicological impacts from the use of the grenade. Also for the Ecoindicator the toxicological impacts dominate although the use of fossil fuels is also important for the peace scenario. For the EPS system on the other hand, the results are dominated by the use of abiotic resources.

4 Discussion

A problem when evaluating weighting methods is that we don't know which values and which results are the right ones. But we can compare the results with our intuitive understanding. An advantage with a monetary method is that the units are understandable. Everybody can look at the weighting factors in Tables 1 and 2 and make their own judgements concerning the reasonableness of the factors. This is more difficult if the weighting factors are expressed in some dimensionless units.

We can also try to evaluate the reasonableness of the results of the case studies. In the case study on waste management, the Ecotax method suggests that toxicological impacts, resource issues and climate change are the most important environmental problems associated with the Swedish waste management system. This seems to mirror the public debates. Also for the agricultural production system, it is interesting to note that the Ecotax method could capture the eutrophication problem which is much discussed. It is interesting to note that the Ecotax method is able to identify different environmental problems as the most important ones in different case studies. This is partly in contrast to other weighting methods. For example, in the EPS system it seems like climate change and the use of abiotic resources are always the most important impacts. For the Ecoindicator it seems like resources, and the traditional inorganic air pollutants are most often the most important ones, sometimes supplemented with metal emissions and climate change.

Some checks of internal consistency can also be made. It was noted above that the NO_x-fee (40 SEK/kg) was not used as a basis for calculation. We can however compare this tax with the calculated valuation from the Ecotax method by calculating the valuation weighting factor for NO_x for the impact categories eutrophication, acidification, human tox and photochemical oxidation. When these are added the sum is 16 SEK/kg and 28 SEK/kg for the min and max version respectively. The values are thus slightly lower than the tax but in the same order of magnitude.

Another possibility to evaluate the weighting factors is to compare the monetary measures with other monetised valuation factors. However, when doing that, the discussion in the introduction on the different types of economic values should be remembered. The taxes can for example be compared to the values calculated in the ExternE project which used a damage cost approach. The Swedish values for NO_x was 18.5 SEK/kg and for SO_2 22.5 SEK/kg (Nilsson and Gullberg 1998) which are a little bit lower than the taxes used (taxes and fees used: 40 SEK/kg and 30 SEK/kg respec-

tively, but in the same order of magnitude). The NO_x value is good accordance with the NO_x valuation in the Ecotax method (16–28 SEK/kg). For CO_2, estimations of external costs vary a lot depending on the choice of the discount rate and function and impacts considered (c.f. Krewitt 2002, Azar and Sterner 1996).

There are a number of characterisation methods available (c.f. Pennington et al. 2004). There is currently no consensus on a specific set or on criteria for choosing. In this version we chose the base line set of the Dutch guideline on LCA (Guinée 2002) as implemented in the Simapro software as a basis since it is a fairly recent and consistent set of European characterisation methods at mid-point level developed in project involving review processes. However for the abiotic resource impact category we chose the thermodynamic approach instead (Finnveden and Östlund 1997). This method describes the resource consumption as consumption of exergy or production of entropy. Exergy can be seen as a measure of the available energy and entropy is defined by the second law of thermodynamics. This approach is suggested to give a scientific definition of the consumption of resources (ibid).

It is however important to note that the weighting method as such is independent of the chosen characterisation methods. The weighting factors in Table 1 can in principle be combined with any characterisation methods. Parallel characterisation methods may also be used giving indications of the uncertainty in the natural scientific basis of the impact assessment methodology (Johansson 1999). An interesting further development is thus to combine the weighting factors with other characterisation methods. The method should also be further developed with respect to exotaxes and fees in other countries as well. Another possible further development is to consider costs which are associated with regulations of emissions. For some pollutants these costs may be better expressions of the societal values.

An obvious problem with the Ecotax method is that monetary measures which are decided for one purpose (taxation) are assumed to be valid for another purpose, in this case as a measure of the environmental importance. We don't know if this assumption is correct. It depends on what the basis for the decision on the tax levels was. We can get some information on this by studying the background documents that were used when the tax decisions were taken. In some cases the tax is based on the environmental objective for the particular pollutant and information about abatement costs. The tax can then be put to a level which means that the tax equals the marginal costs of reducing the emissions to the desired level. If the tax level corresponds to the external costs, the result will be that the level of the emissions is on an optimal level according to economic theory (Turner et al. 1994). If this optimal level of emission is the same as the environmental objective, than the marginal cost of reaching the objective will correspond to the external cost. In other cases, the tax level is gradually increased as long as the emissions level is judged to be higher than optimal. However, when deciding on tax levels, decision makers will also take other concerns. For example, they may not be prepared to jeopardise the competitiveness of important industrial sectors by imposing high taxes on them. This type of concern may result in different types of tax exemptions and subsidies, and this is one reason why we choose to always use the highest tax level for a specific pollutant and regard the other levels as subsidies. Even if the tax does not correspond to the external costs, or a reasonable approximation of them, they may still be used as a basis for weighting factors, if the deviation from the external costs is relatively constant by a certain factor.

A key aspect when evaluating the Ecotax method is whether it can be assumed that the taxes correspond to the external costs, or at least reasonable approximations of them. Unfortunately we do not know the answer to this question, partly because we do not know which are the 'right' external costs. A certain leap of faith is thus necessary. The Ecotax approach is based on a positive view on representative democracy implying that governments make decisions from which meaningful information can be drawn (Finnveden 1997).

The Ecotax approach avoids some problems which are more or less inherent in other approaches. A key problem with methods based on stated preferences is that values are created during the elicitation process. This means that the results are sensitive to the process and different types of biases may occur (Finnveden et al. 2002, Mettier and Hofstetter 2005). These types of problems are avoided by using a revealed preferences method.

By using a mid-point method, a number of problems associated with damage methods are avoided. For example, there is no need to model the cause-effect chain all the way to the damage level, thus avoiding different types of uncertainties associated with such modelling. Furthermore, the question of discounting which is necessary in damage modelling and valuation is avoided here. Because decision-makers may have considered unknown effects and the precautionary principle when they decided on tax levels, such aspects may have been considered in the method, in contrast to damage methods which can only consider known and quantifiable effects.

The midpoint-endpoint debate has often centred on different types of uncertainties (e.g. Bare et al. 2000, Hertwich and Hammit 2001, Lanzen 2005). Sometimes it is claimed that an advantage with having an endpoint approach is that the weighting would be easier and less uncertain. Here we are however suggesting a mid-point weighting methods that we claim are no less uncertain than other often used weighting methods based on a damage assessment. This paper can therefore be seen as a discussion paper also in the midpoint-endpoint debate.

We have here presented a weighting method based on ecotaxes and fees. The method has been used by us in a number of case studies both for LCAs (e.g. Finnveden et al. 2005, Hochschorner et al. 2005), in the context of SEA (Nilsson et al. 2005) and for sector assessments based on environmentally extended input-output analysis (Engström et al. 2005). Others have also used it in the context of LCC (Carlsson Reich 2005) and CBA (Bäckman et al. 2001). This suggests that the method is possible to use not only for LCA but also for a number of other environmental systems analysis tools.

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